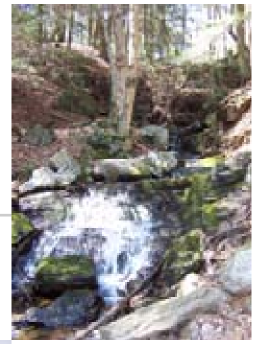
# Ecology of Dead Wood in the Northeast



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forest GUILD

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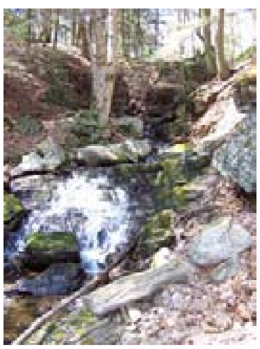

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## *1. Introduction*

Although dead wood and decaying trees have historically had little commercial value, they do have substantial ecological value. This paper reviews the scientific literature to provide the background necessary to craft recommendations about the amount and type of dead wood that should be retained in the forest types of the northeastern U.S. Establishing the ecological requirements for dead wood and other previously low-value material is important because of the increased interest in this material for energy and fuel. More intensive extraction of biomass from the forest may impinge on the forest's ability to support wildlife, provide clean water, sequester carbon, and regenerate a diverse suite of plants.

This background paper covers the topics of dead wood, soil compaction, nutrient conservation, and wildlife habitat in temperate forests generally as well as in specific forest types of the Northeast. Complex issues related to carbon storage in forests and the climate impacts of using forest material for energy and fuel are very important and deserve an in-depth investigation beyond the scope of this paper. Similarly, this paper will not discuss the state of biomass harvesting in the U.S. (Evans 2008, Evans and Finkral 2009) or existing biomass harvesting guidelines (Evans and Perschel 2009) which have been addressed in other recent publications.

The goal of this background paper is to provide a concise summary that can inform discussions about biomass harvesting standards in the Northeast. However, it is important to note that this document makes no suggestions about how a biomass harvest should be conducted or what should be left in the forest after a harvest. Rather we have attempted to lay out the basic science on which recommendations can be built.

## *2. Ecology of Dead Wood in the Northeast*

### 2a. Dead Wood and Stand Development

Dead wood is important not only in terms of total volume or mass in a stand, but also in terms of piece size — usually measured as diameter at breast height (DBH) for snags (and for live trees) or diameter of the large end for down woody material (DWM). Large-diameter snags or down logs are important habitat for numerous animal species, persist for long periods, store nutrients, and provide substrate for seed germination.

The process of dead wood accumulation in a forest stand consists of the shift from live tree to snag to DWM unless a disturbance has felled live trees, shifting them directly to DWM. The graphs below (Figures 1, 2, and 3) show the general pattern of the production of dead wood in total amount and size. The data in these graphs are taken from research in northern hardwood forests (Gore and Patterson 1986, Goodburn and Lorimer 1998, Hale et al. 1999, McGee et al. 1999, Nyland et al. 2000). The 4 in (10 cm) diameter size is within the range of the minimum size used in most coarse woody material (CWM) inventories. Fine woody material (FWM) refers to smaller-sized dead material. The graphs depict the patterns for a stand that had been harvested as a conventional clearcut, leaving a large amount of small woody material (nearly all <10 in (25 cm) diameter), but no trees >4 in (>10 cm) DBH and no snags. The pattern is shown from just after the clearcut (age 0) – age 100 years, and in the old-growth condition.

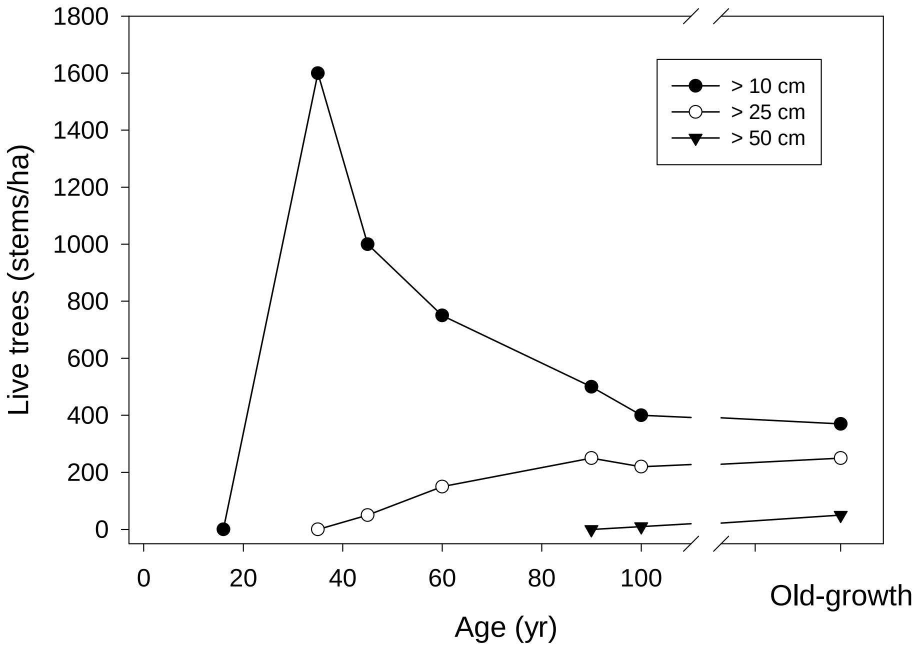


**Figure 1 General Pattern of Tree Density Over Time**

The young stand produces large numbers of trees (~600 stems/ac or ~1500 stems/ha) at age 30, and the intense competition among these trees causes mortality of smaller stems, which creates an increasing number of small snags (Figure 2). Trees begin to grow into 10 in (25 cm) DBH size by age 40, and trees of this size begin to dominate the stand by age 80. Snags of the 10 in (25 cm) DBH size begin to appear at age 60 as mortality of larger trees occur. Large live trees (>20 in or >50 cm) begin to appear at age 90 – 100, with snags of that size as well.

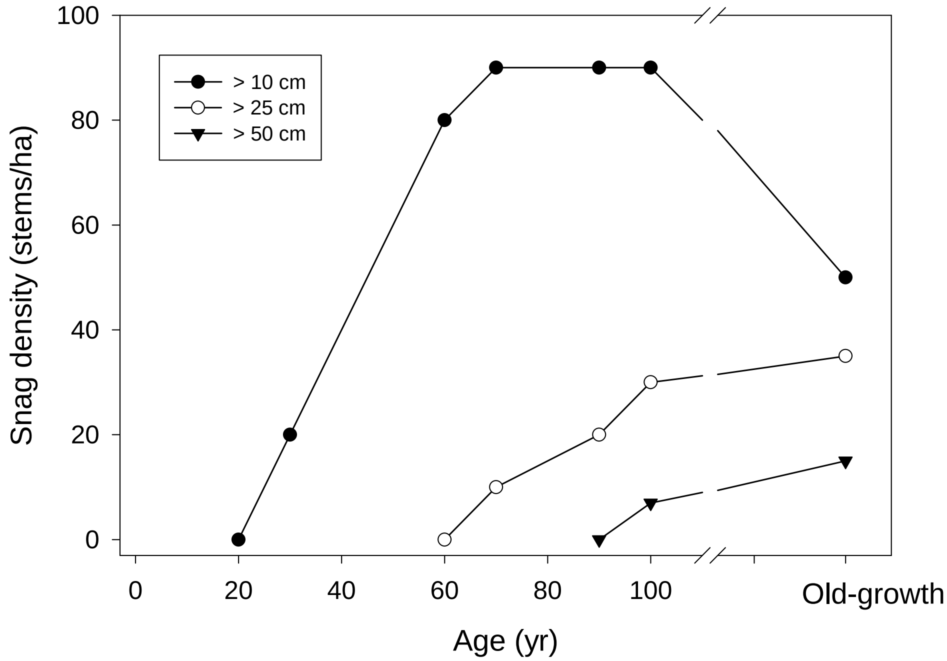


**Figure 2 General Pattern of Snag Density Over Time**

The large amount of DWM present just after the clearcut (which consists mostly of pieces <10 in (<25 cm) diameter) decomposes rapidly in the first 25 years and continues to decline in mass to age 40. From age 40 – 100 years, DWM increases as small snags fall, and then larger snags begin to contribute to DWM that include pieces >10 in (>25 cm) diameter. Very few large (> 20 in or >50 cm) pieces of DWM are produced. Large DWM often results from wind or other disturbances that fell large trees in the old-growth stage. Thus, large DWM tends to accumulate periodically from these disturbance pulses; whereas small DWM accumulates in a more predictable pattern in earlier stages of stand development.

This process produces the U-shaped pattern that is often described with a dearth of DWM in the intermediate ages (Figure 3). This pattern shows the importance of retaining large live trees and large snags at the time of harvest; they will contribute large DWM to the forest floor throughout the development of the stand.

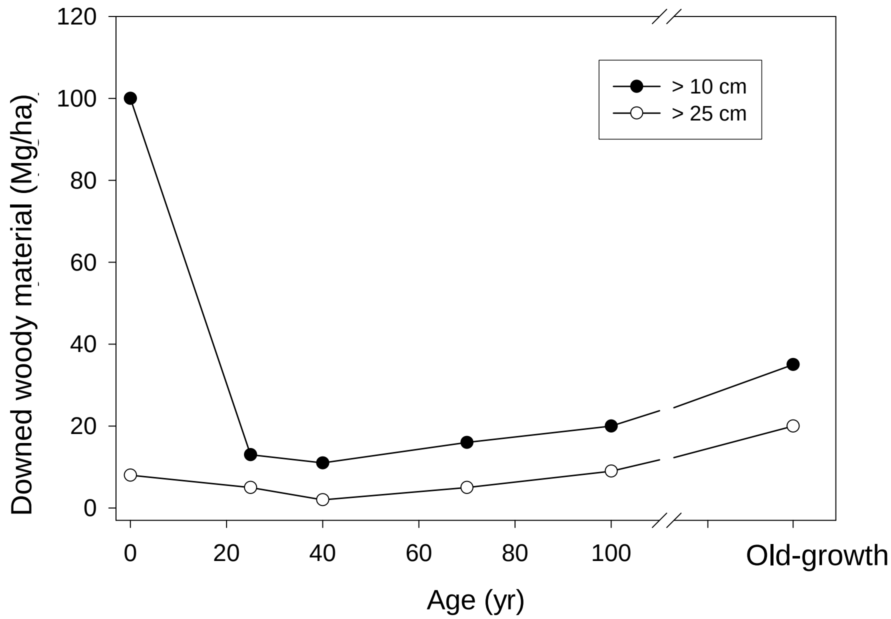


**Figure 3 General Pattern of DWM Density Over Time**

## 2b. Wildlife and Biodiversity

Dead wood is a central element of wildlife habitat in forests (Freedman et al. 1996). Many forest floor vertebrates have benefited or depended on DWM (Butts and McComb 2000). In the southeastern U.S., more than 55 mammal species, over 20 bird species, and many reptiles and amphibian species were relying on dead wood (Lanham and Guynn 1996, Loeb 1996, Whiles and Grubaugh 1996) with similar numbers for the forests of the Pacific Northwest (Carey and Johnson 1995, McComb 2003). In New England, De Graaf and colleagues (1992) catalogued at least 40 species that rely on DWM.

Some examples of relationships between animals and DWM in the Northeast include a study showing that low densities of highly decayed logs (less than one highly decayed log/ha ) had a negative impact on red-back voles (*Clethrionomys gapperi*) in a northern hardwood forest in New Brunswick, Canada (Bowman et al. 2000). DWM retention increased spotted salamander (*Ambystoma maculatum*) populations in a Maine study (Patrick et al. 2006). While DWM is important habitat for red-backed voles in Maine, it did not effect populations at volumes as low as 543 $ft^3$/ac (38 $m^3$/ha; McCay and Komoroski 2004). The quantity of DWM had no effect on white-footed mice (*Peromyscus leucopus*) abundance in an Appalachian study, but at the micro-site scale, mice were more often located near DWM (Marcus et al. 2002). Similarly, shrew (*Tupaia* sp.) showed minimal or no response to drastic decreases in the abundance of large logs in managed loblolly pine (*Pinus taeda*) forests of the southeastern coastal plain (McCay and Komoroski 2004) .

In aquatic environments, DWM provided crucial refuge from predation (Angermeier and Karr 1984, Everett and Ruiz 1993). Logs that fell in the water formed a critical component of aquatic habitat by ponding water, aerating streams, and storing sediments (Gurnell et al. 1995, Sass

2009). In fact, removal of large woody material from streams and rivers had an overwhelming and detrimental effect on salmonids (Mellina and Hinch 2009).

DWM is a key element in maintaining habitat for saproxylic insects (Grove 2002). For example, some specialist litter-dwelling fauna that depend on DWM appear to have been extirpated from some managed forests (Kappes et al. 2009). A study from Ontario suggests that overall insect abundance was not correlated with the volume of DWM, though abundance of the fungivorous insect guild was positively related to the volume of DWM (Vanderwel et al. 2006b). Extensive removal of DWM could reduce species richness of ground-active beetles at a local scale (Gunnarsson et al. 2004). More generally, a minimum of 286 $ft^3$/ac (20 $m^3$/ha) of DWM has been suggested to protect litter-dwelling fauna in Europe (Kappes et al. 2009).

Dead logs served as a seedbed for tree and plant species (McGee 2001, Weaver et al. 2009). Slash could be beneficial to seedling regeneration after harvest (Grisez, McInnis, and Roberts 1994). Fungi, mosses, and liverworts depended on dead wood for nutrients and moisture, and in turn, many trees were reliant on mutualistic relationships with ectomycorrhizal fungi (Hagan and Grove 1999, Åström et al. 2005). In general, small trees and branches hosted more species of fungus per volume unit than larger trees and logs; however larger dead logs may be necessary to ensure the survival of specialized fungus species such as heart-rot agents (Kruys and Jonsson 1999, Bate et al. 2004).

### 2c. Soil Productivity

DWM plays an important physical role in forests and riparian systems. DWM added to the erosion protection by reducing overland flow (McIver and Starr 2001, Jia-bing et al. 2005). DWM also had substantial water-holding capacity (Fraver et al. 2002). DWM in riparian systems provided sites for vegetation colonization, forest island growth and coalescence, and forest floodplain development (Fetherston et al. 1995).

In many ecosystems, CWM decomposed much more slowly than foliage and FWM, making it a long-term source of nutrients (Harmon et al. 1986, Johnson and Curtis 2001, Greenberg 2002, Mahendrappa et al. 2006). DWM decomposed through physical breakdown and biological decomposition (Harmon et al. 1986). The diameter of each piece of DWM, temperature of the site, amount of precipitation, and tree species all influenced the rate of DWM decomposition (Zell et al. 2009). In general, conifers decayed more slowly than deciduous species (Zell et al. 2009). Other factors that encouraged decomposition included warmer temperatures, rainfall between 43 and 51 in/year (1100 and 1300 mm/year), and small-sized pieces (Zell et al. 2009). While there is great variation across ecosystems and individual pieces of DWM, log fragmentation generally appears to occur over 25 – 85 years in the U.S. (Harmon et al. 1986, Ganjegunte et al. 2004, Campbell and Laroque 2007).

In some ecosystems, DWM represents a large pool of nutrients and is an important contributor to soil organic material (Graham and Cromack Jr. 1982, Harvey et al. 1987). However, review of DWM in Northern coniferous forests suggested that DWM may play a small role in nutrient cycling in those forests (Laiho and Prescott 2004). The same review showed that DWM contributes less than 10 percent of the nutrients (Nitrogen (N), Phosphorus (P), Potassium (K), Calcium (Ca), and Magnesium (Mg) returned in aboveground litter annually, and approximately five percent of the N and P released from decomposing litter or soil annually (Laiho and Prescott 2004). Although DWM is often low in N itself, N fixation in DWM was an important source of this limiting nutrient in both terrestrial and aquatic ecosystems (Harmon et al. 1986). There was a wide range of non-symbiotic N fixation, but temperate forests received average input of about 1.8 – 2.7 lb/ac/yr (2 – 3 kg/ha/year) of N (Roskoski 1980, Yowhan Son 2001).

A review of scientific data suggests that when both sensitive sites (including low-nutrient) and clearcutting with whole-tree removal are avoided, then nutrient capital can be protected (see also Hacker 2005).  However, there is no scientific consensus on this point because of the range of treatments and experimental sites (Grigal 2000). It is important to emphasize that the impact on soil nutrients is site dependent. Low-nutrient sites are much more likely to be damaged by intensive biomass removal than sites with great nutrient capital or more rapid nutrient inputs. A report on impacts of biomass harvesting from Massachusetts suggested that with partial removals (i.e., a combination of crown thinning and low thinning that removes all small trees for biomass and generates from 9 – 25 dry t/ac or 20 – 56 Mg/ha) stocks of Ca, the nutrient of greatest concern, could be replenished in 71 years (Kelty et al. 2008). The Massachusetts study was based on previous research with similar results from Connecticut (Tritton et al. 1987, Hornbeck et al. 1990). Leaching, particularly of Ca due to acidic precipitation, can reduce the nutrients available to forests even without harvests (Pierce et al. 1993). However, the Ca-P mineral apatite may provide more sustainable supplies of Ca to forests growing in young soils formed in granitoid parent materials (Yanai et al. 2005).

15 years of data from Hubbard Brook Ecosystem Study indicate that a whole-tree clear cut did not result in the depletion of exchangeable Ca pools (Campbell et al. 2007). The Environmental Impact Statement from the White Mountain National Forest (2005 p. 3 – 19) demonstrated the variation in Ca removed by treatment and forest type, though even whole-tree clear cut was estimated to remove only four percent of the total Ca pool. A study of an aspen/mixed-hardwood forest showed that even with a clearcut system, Ca stocks would be replenished in 54 years (Boyle et al. 1973). Minnesota's biomass guidelines present data that showed soil nutrient capital was replenished in less than 50 years even under a whole-tree harvesting scenario (Grigal 2004, MFRC 2007). Whole-tree clearcutting (or whole-tree thinning, e.g., Nord-Larsen 2002) did not greatly reduced amounts of soil carbon or N in some studies (Hornbeck et al. 1986, Hendrickson 1988, Huntington and Ryan 1990, Lynch and Corbett 1991, Olsson et al. 1996, Johnson and Todd 1998). Lack of significant reduction in carbon and N may be due to soil mixing by harvesting equipment (Huntington and Ryan 1990). However, intensive cutting, such as

clearcutting with whole-tree removal, can result in significant nutrient losses (Hendrickson 1988, Federer et al. 1989, Hornbeck et al. 1990, Martin et al. 2000, Watmough and Dillon 2003)—in one case, 13 percent of Ca site capital (Tritton et al. 1987).

Low-impact logging techniques that reduce soil disturbance can help protect nutrient capital (Hallett and Hornbeck 2000). Harvesting during the winter after leaf fall can reduce nutrient loss from 10 – 20 percent (Boyle et al. 1973, Hallett and Hornbeck 2000). Alternatively, if logging occurs during spring or summer, leaving tree tops on site would aid in nutrient conservation. Nordic countries have demonstrated that leaving cut trees on the ground in the harvest area until their needles have dropped (one growing season) can also reduce nutrient loss (Nord-Larsen 2002, Richardson et al. 2002).

### 2d. Quantities of Dead Wood

Site productivity and the rate of decomposition helped determine the amount of dead wood in a given stand (Campbell and Laroque 2007, Brin et al. 2008). As mentioned above, DWM decomposition varies greatly but generally occurs over 25 – 85 years in the U.S. (Harmon et al. 1986, Ganjegunte et al. 2004, Campbell and Laroque 2007). All mortality agents including wind, ice, fire, drought, disease, insects, competition, and senescence create dead wood (Jia-bing et al. 2005). Of course, these mortality agents often act synergistically.

A review of 21 reports of quantitative measures of DWM in Eastern forest types shows great variability across forest types and stand development stages (Roskoski 1980, Gore and Patterson 1986, Mattson et al. 1987, McCarthy and Bailey 1994, Duvall and Grigal 1999, Idol et al. 2001, Currie and Nadelhoffer 2002). The reports ranged from 3 – 61 t/ac (7 to 137 Mg/ha) with a median of 11 t/ac (24 Mg/ha) and a mean of 15 t/ac (33 Mg/ha; see Figure 4). Measurements of old forests (>80 years old), had a median of 11 t/ac (24 Mg/ha) and a mean of 13 t/ac (29 Mg/ha) in DWM.

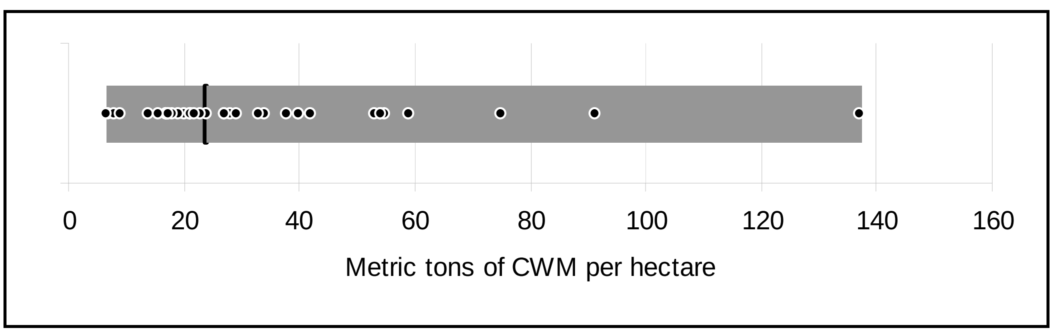


**Figure 4 Distribution of DWM Measured in Eastern Forests**

The gray bar shows the range of DWM measurement, the black line shows the median value, and each dot represents one measurement of DWM.

In contrast, a study of U.S. Forest Service inventory plots found a mean of 3.7 t/ac (8.3 Mg/ha) and a median of 2.9 t/ac (6.5 Mg/ha) of DWM across 229 plots in the Northeast (Chojnacky et al. 2004 see Figure 2). This low level of DWM across the landscape may be due to widespread clearcutting in the 1880-1930 period.

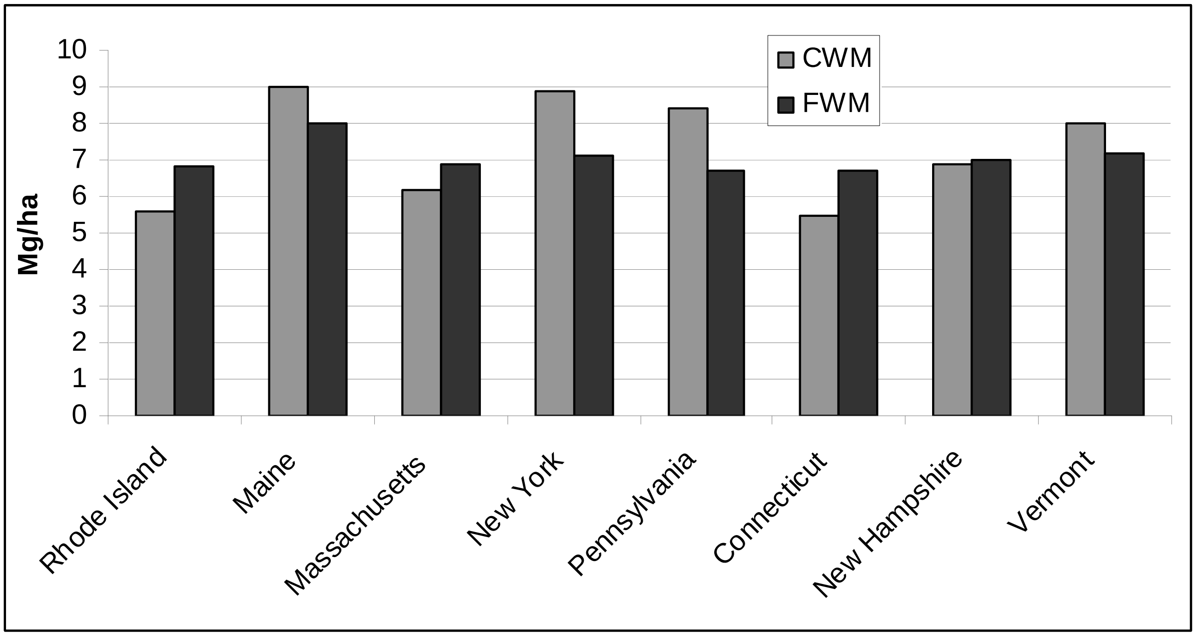


**Figure 5 U.S. Forest Service Inventory Estimates of Deadwood**
**Data from Chojnacky et al. 2004**

## *3. Research by Forest Type*

The following section uses the best available scientific literature to examine the dead wood dynamics of specific forest types in the Northeast. This region encompasses three ecological provinces including Northeastern mixed forest, Adirondack-New England mixed forest-coniferous forest, and Eastern broadleaf forest (McNab et al. 2007). Major forest types in the region are white/red/jack pine (*Pinus sp.*), spruce-fir (*Picea sp. - Abies sp.*), oak-hickory (*Quercus sp. - Carya sp.*) or transitional hardwood forests, and northern hardwood forests (Eyre 1980).

The average year round temperature in the Northeast is 46°F (8°C). Winter temperatures average 24°F (-4.3°C) while summer temperatures average 67°F (19.6°C; National Climate Data Center 2008). The prevailing wind direction, from west-to-east, creates a continental climate except for coastal areas moderated by the Atlantic Ocean (Barrett 1980). On average, the region receives 41 in (104 cm) of precipitation which is evenly distributed throughout the year (National Climate Data Center 2008). Elevations range from sea level to mountain tops above 5,300 ft (1,600 m), but much of the region is set on upland plateaus between 500 ft and 1500 ft (150 and 460 m; Barrett 1980). Glaciation created young soils which vary considerably across small spatial scales (Barrett 1980).

Much of the southern portion of Northeastern forests was cleared for agriculture in the early 19th century, leaving less than one percent of the forest cover in an old-growth condition (Cogbill et al. 2002). Currently much of the region is comprised of second- or third-growth forest that has yet to reach late seral stages (Irland 1999). There are about 80 million ac (32 million ha) of timberlands (areas where commercial timber could be produced) and about 4 million ac (1.6 million ha) of reserved forest where harvests are not permitted (Alvarez 2007). Approximately 1,272 million $ft^3$ (36 million $m^3$) of wood are harvested annually out of 3,157 million $ft^3$ (89 million $m^3$) of net tree growth (Alvarez 2007).

### 3a. Spruce-Fir Forests

Spruce-fir forests dominate the inland areas of Maine as well as the mountain tops northernmost portions of New York, New Hampshire, and Vermont. These forests have cold temperatures and relatively coarse, acidic soils (Barrett 1980). Dead wood is important in spruce-fir ecosystems. For example, in Maine (the state with the greatest area of spruce-fir forests in the Northeast), DWM, snags, and cavity trees are important habitat for 20 percent of bird, 50 percent of mammal, 44 percent of amphibian, and 58 percent of reptile species found there (Flatebo et al. 1999). Animals that rely on DWM in spruce-fir forests include pine marten (*Martes americana atrata*) (Kyle and Strobeck 2003) and may include some saproxylic vertebrates (Majka and Pollock 2006).

In 2001, researchers found the volume of down dead wood in Maine's spruce-fir forest to be 530 $ft^3$/ac (37 $m^3$/ha) or 3.4 t/ac (7.5 Mg/ha) (Heath and Chojnacky 2001, Table 36). While the average was 3.4 t/ac (7.5 Mg/ha) non-industrial private lands only had 3 t/ac, public lands had 3.3 t/ac, while industrial lands had 3.7 t/ac (Heath and Chojnacky 2001, Table 37). The quadratic-mean, large-end diameter of down wood in Maine's spruce fir-forests measured 6.7 in (17 cm; Heath and Chojnacky 2001). The number of dead trees in nine red spruce-balsam fir forests ranged from 85 – 232/ ac (210 – 574/ ha) or from 11 – 43 percent of the basal area (Tritton and Siccama 1990). The nine paper birch-red spruce-balsam fir stands survey ranged from 33-86 dead trees/ac (81 to – 212/ha) or 11 – 35 percent of basal area (Tritton and Siccama 1990), and overall, 14 percent of the trees in Maine were standing dead (Griffith and Alerich 1996). Dead wood provided an important substrate for spruce and hemlock seedling development (Weaver et al. 2009). While a commercial clearcut reduced the area of dead wood available for seedling growth, 5- and 20-year-selection cutting cycles were not statistically different from the uncut reference stand with 362 – 501 $ft^2$/ac (83 – 115 $m^2$/ha) of dead wood (Weaver et al. 2009).

As described above, spruce-fir forests tend to have two peaks in DWM over time: one early in stand development and a second peak after the stem exclusion phase (Figure 3). For example, one study showed a change from 63 t/ac (28 Mg/ha) in a stand <20 years, 22 t/ac (10 Mg/ha) in the 41 – 60-year age class, to 117 t/ac (52 Mg/ha) in the 61 – 80-year age class, and returning to less than 56 (25 Mg/ha) in the 101 – 120-year age class (Taylor et al. 2007). Fraver and colleagues (2002) showed that pre-harvest an Acadian forest had 10 t/ac (23 Mg/ha) of DWM. The harvest in this study increased the mass of DWM, but more of the pieces were small diameter (Fraver et al. 2002). While the harvest method (whole tree, tree length, or cut to length) and harvest system affect the amount of DWM left after harvest, many studies do not specify how material was removed.

Snag densities in balsam fir forests of Newfoundland followed a similar pattern over time. Stands contained nearly 16 snags/ac (40/ha) the first year post harvest; then the density declined below the 4 snags/ac (10/ha) required by the regional forest management guidelines at 20 years post harvest; and finally the number of snags returned to initial levels in the 80 – 100 years post-harvest stands (Smith et al. 2009). Smith and colleagues (2009) recommended retention and recruitment of white birch snags to ensure sufficient snag and DWM density. The Canadian province of Newfoundland and Labrador requires retention of 4 snags/acre while Maine recommends retention of 3 snags greater than 14 inches DBH and one greater than 24 inches DBH (Flatebo et al. 1999, Smith et al. 2009). Other guidelines recommend between 5 and 6 snags/acre greater than 8 inches and an additional 4 – 6 potential cavity trees (Woodley 2005).

A study of two old-growth balsam and black spruce sites demonstrated a wide range of average DWM piece sizes even in unmanaged lands. In the two study sites, the average diameter of the DWM structures were 54.8 cm and 16.1 cm; average height of snags was 4.73 m and 2.52 m; and length of logs were 5.91 m and 4.81 m (Campbell and Laroque 2007). The differences between the two sites are due, in part, to differences in rates of decomposition, i.e., higher rates of decomposition reduce the average size of DWM pieces.

One study of pre-commercial thinning in spruce-fir forests showed that the mass of DWM was reduced from 29 – 15 t/ac (64 – 34 Mg/ha; Briggs et al. 2000). In one study of a spruce-fir whole tree clearcut in Maine, 35 percent of organic matter was in trees and 12 percent was in woody litter and forest floor (Smith Jr et al. 1986). In that study, 23 t/ac (52 Mg/ha) of DWM were left after the harvest, but the whole-tree removal took about 91 percent of N, P, K, and Ca from the site, which was between 2 and 4 times the nutrient removal from a bole-only harvest (Smith Jr et al. 1986). Depletion of Ca is of some concern in Maine, though not as great a concern as in the Central and Southeastern U.S. (Huntington 2005). Spruce-fir forests generally incorporate Ca into merchantable wood at 1.6 kg Ca/ac/yr (1.6 kg $ha^{-1}yr^{-1}$; Huntington 2005). Some sites such as Weymouth Point, Maine, have documented Ca-depletion problems (Smith Jr et al. 1986, Hornbeck et al. 1990, Briggs et al. 2000). The rate of weathering replenishment of Ca in Maine is uncertain, and the Ca-rich mineral apatite may be an important source of Ca (Huntington 2005, Yanai et al. 2005). Climate change and the associated warming and species composition shift may exacerbate Ca depletion in spruce-fir forests (Huntington 2005).

### 3b. Northern Hardwood Forests

Northern hardwood forests are dominated by maple (*Acer sp.*), beech (*Fagus grandifolia)*, and birch (*Betula sp.*) and cover lower elevations and southern portions of Maine, New York, New Hampshire, Vermont, and the northern portion of Pennsylvania. Northern hardwood forests also include conifers, e.g., hemlock (*Tsuga canadensis*) and white pine (*Pinus strobus*), in the mixture (Westveld 1956).

In general, the amount of DWM in northern hardwood forests follows the 'U' pattern mentioned above. Young stands have large quantities of DWM; mature stand have less; and older or uncut stands have more. For example, a study in New Hampshire measured 38 t/ac (86 Mg/ha) of DWM in a young stand, 14 t/ac (32 Mg/ha) in mature stands, 20 t/ac (54 Mg/ha) in old stand, and 19 t/ac (42 Mg/ha) in an uncut stand (Gore and Patterson 1986). Gore and Patterson (1986) also note that stands under a selection system had lower quantities of DWM, i.e., 16 t/ac (35 Mg/ha). A review of other studies identified similar temporal patterns and quantities of DWM (see Figure 6 from data described in Roskoski 1977, Tritton 1980, Gore and Patterson 1986, McCarthy and Bailey 1994, McGee et al. 1999, Bradford et al. 2009).

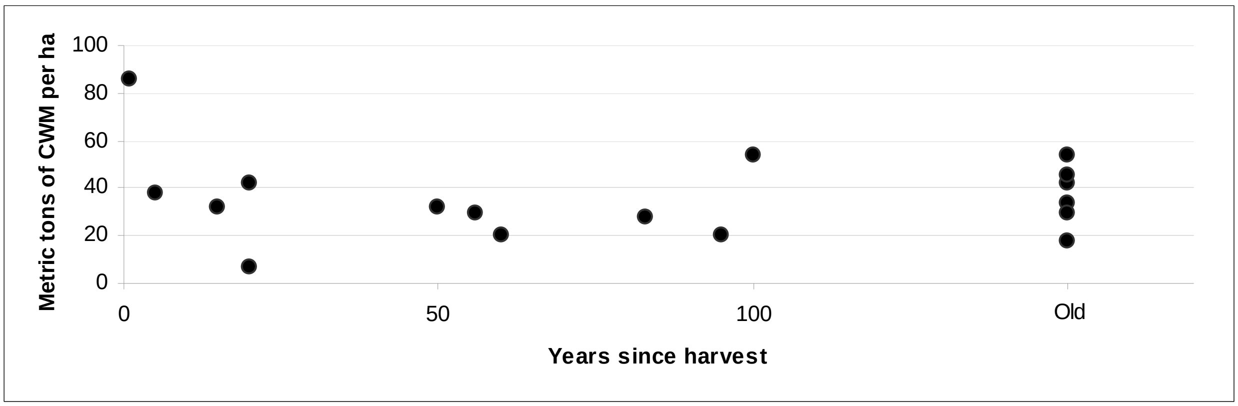


**Figure 6 Quantities of DWM in Northern hardwood forests Forests**
**Data described in Gore and Patterson 1986, McCarthy and Bailey 1994, McGee et al. 1999, Bradford et al. 2009, and Roskoski 1977**

Estimates of the volume of down dead wood in Maine's northern hardwood forests are 598 $ft^3$/ac (42 $m^3$/ha) or 9 t/ac (20.5 Mg/ha Heath and Chojnacky 2001). Keeton (2006) estimates a volume of 600 $ft^3$/ac (42 $m^3$/ha) of DWM in a multi-aged northern hardwood forest.

The number of dead trees in five hemlock-yellow birch forests range from 16 – 45/ac (40 – 112/ha) or from 3 – 14 percent of the basal area (Tritton and Siccama 1990). The 14 sugar maple-beech-yellow birch stands survey ranged from 14 – 99 dead trees/ac (35 – 245/ha) or 5 – 34 percent of basal area (Tritton and Siccama 1990). Other estimates of snag densities in northern hardwood forests include 5/ac (11/ha) (Kenefic and Nyland 2007), 15/ac (38/ha) (Goodburn and Lorimer 1998), and 17/ac (43/ha) (McGee et al. 1999). Tubbs and colleagues (1987) recommend leaving a between of one and ten live decaying trees/acre of least 18 inches DBH.

The number of cavity trees is another important habitat element in northern hardwood forests that is reduced by harvest. For example, studies in northern hardwood forests have shown a reduction from 25 cavity trees/ac (62/ha) before harvest and to 11 (27/ha) afterward (Kenefic and Nyland 2007). Another study measured 7 cavity trees/ac (18/ha) in old-growth, 4/ac (11/ha) in even-aged stand, and 5/ac (13/ha) in a stand selection system (Goodburn and Lorimer 1998).

### 3c. Transition Hardwood Forests

Oak-hickory forests occupy the southernmost portions of the region. The oak-hickory forests are also considered a transitional forest type between the northern hardwood forests type and the Appalachian hardwoods that dominate further south (Westveld 1956).

As with the other forest types discussed, DWM density tends to follow a 'U' shape in oak-hickory forests. For example, Idol and colleagues (2001) found 61 t/ac (137 Mg/ha) in a one-year post-harvest stand, 18 t/ac (40 Mg/ha) in a 31-year-old stand, and 26 t/ac (59 Mg/ha) in a 100-year-old stand. Tritton and colleagues (1987) measured 5.8 t/ac (13 Mg/ha) in an 80-year-old stand in Connecticut.

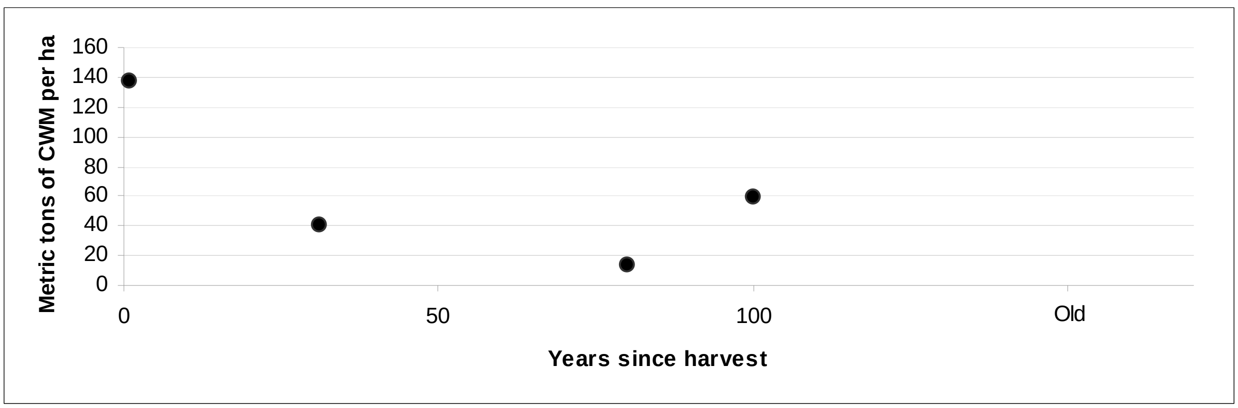


**Figure 7 DWM in Oak-Hickory Forests**
**Data described in (Tritton et al. 1987, Idol et al. 2001)**

Estimates of the volume of down dead wood in Maine's oak-hickory forests are 244 $ft^3$/ac (17 $m^3$/ha) or 0.7 (1.5 Mg/ha; Heath and Chojnacky 2001). Wilson and McComb (2005) estimated the volume of downed logs in a western Massachusetts forest at 143 $ft^3$/ac (10 $m^3$/ha).

Out of seven oak stands in Connecticut, the number of dead trees ranged from 19 – 44/ac (46 – 109/ha) or 5 – 15 percent of basal area (Tritton and Siccama 1990). The decadal fall rates of snags in a Massachusetts study varied from 52 – 82 percent (Wilson and McComb 2005). Snags, particularly large-diameter snags, provide important nesting and foraging sites for birds (Brawn et al. 1982). In general, wildlife habitat requirements for dead wood are poorly documented, but it is clear that some wildlife species rely on dead wood in oak-hickory forests (Kluyver 1961, DeGraaf et al. 1992).

A study in Appalachian oak-hickory forests showed that the decomposing residues left after a sawlog harvest increased concentration of Ca, K, and Mg in foliage and soils after 15 years in comparison to a whole-tree harvest (Johnson and Todd 1998). However, the study found no impacts on soil carbon, vegetation biomass, species composition, vegetation N or P concentration, soil-bulk density, or soil N because of the whole-tree harvest (Johnson and Todd 1998).

### 3d. White Pine and Red Pine Forests

Pine forests are found in the coastal areas of Maine and New Hampshire and much of central Massachusetts. Pine forests tend to occupy sites with coarse-textured, well-drained soils (Barrett 1980).

Estimates of the volume of down dead wood in Maine's pine forests are 255 $ft^3$/ac (18 $m^3$/ha) or 1.6 t/ac (3.5 Mg/ha; Heath and Chojnacky 2001). A review of research on DWM in the red pine forests of the Great Lakes area showed that there were 50 t/ac (113 Mg/ha) of DWM in an unmanaged forest at stand initiation and 4.5 t/ac (10 Mg/ha) in a 90-year-old stand (Duvall and Grigal 1999). In comparison, the managed stand Duvall and Grigal (1999) studied had less DWM at both initiation 8.9 t/ac (20 Mg/ha) and at 90 years 2.9 t/ac (6.6 Mg/ha). The same review showed the unmanaged stand had 30 snags/ac (74/ha) while the managed forest had 6.9/ac (17/ha; Duvall and Grigal 1999). Red and white pine that fall to the ground at time of death will become substantially decayed (decay class IV of V) within 60 years (Vanderwel et al. 2006a).

While not a recognized forest type, stands with a mix of oak, other hardwoods, white pine, and hemlock are common. Many of the red oak and white pine stands on sandy outwash sites are susceptible to nutrient losses because of a combination of low-nutrient capital and past nutrient depletion (Hallett and Hornbeck 2000).

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